CHAPTER 1

INTRODUCTION

1.1 Indoor air pollution

Air pollution is usually a serious problem in cities. Air pollutants include various gases and particulates that can harm human health and damage the environment. They may be gases, liquids or solids. Many pollutants are given off into the air as a result of human behavior. Pollution occurs on different levels: personal, national and global. Pollutants are a major public health concern, which include particulate matters (PMs), carbon dioxide (CO_2), carbon monoxide (CO), nitrogen oxides (NO_X), volatile organic compounds (VOCs), black carbon (BC) and sulfur dioxide (SO_2) (Defra, 2010). Air pollution has become extensive problem in several countries. It has caused major concern over the world due to its widespread nature, damaging our environment and is a potential health risk to humans (Navasumrit et al., 2008; Jia et al., 2011; Sarkar and Khillare, 2013).

Air pollution can contaminate both the indoor and outdoor environment, which is a major environmental health problem. Indoor air pollution is caused by cigarette smoking, cooking stove, incense burning, building activity, cleaning products and home furnishings (Wang et al., 2007; Kim et al., 2013). The problem regarding outdoor air pollution has been well publicized for several decades. However, indoor air pollution is causing health impact for obvious reasons.

Many researchers have studied the adverse health effects of indoor air pollution on humans. Indoor pollutants have several sources such as total suspended particulate (TSP), coarse particulate (PM_{10}) and fine particulate ($PM_{2.5}$) and polycyclic aromatic hydrocarbons (PAHs) in particulate emitted from very intensive traffics in New York town and Lithuania (Jung et al., 2010; Krugly et al., 2014), while in Japan and Thailand, $PM_{2.5}$ and PAHs were released from cigarette smoking (Ohura, et al., 2004; Charoenca et al., 2013). Burning of biomass fuel (wood, dung, twig, straw and dry leaf) in stove is one major source of TSP, PM_{10}, $PM_{2.5}$, CO, CO_2, NO_X and PAHs in indoor air (Bruce et

al., 2000; Chakraborty et al., 2014). In Hong Kong and Taiwan, researchers reported incense burning in temples emitted air pollutants such as particle (TSP, PM_{10} and $PM_{2.5}$), VOCs, PAHs, CO, NO_X and methane (CH_4). It was found that $PM_{2.5}$ and PM_{10} concentrations exceeded the Indoor Air Quality Objectives (IAQO) used for office building and public places (Wang et al., 2007; Fang et al., 2002). In Thailand, Navasumrit et al. (2008) studied the characteristics of pollutants emitted from incense burning in temples in Samutprakarn, Chachengsao and Ayutthaya Province. They found benzene, 1,3-Butadiene and PAHs bounded with TSP. Chunram et al. 2007 reported that cooking, smoking and incense burning are major sources of indoor $PM_{2.5}$ in residential and workplace buildings in urban Chiang Mai. Klinmalee et al. (2009) reported that indoor pollutants released from traffic including $PM_{2.5}$, SO_2, NO_2, CO, CO_2 and BTEX, which were found at a shopping mall, university campus in northern Bangkok. Therefore, important sources of indoor pollutants are cooking, incense burning, smoking, emissions from building materials, furnishings and appliances and use of consumer products. Some outdoor activities such as traffic and biomass burning can also influence levels of pollutant indoors.

Most indoor air pollutants have directly affected the respiratory and cardiovascular systems. Several studies reported an association between exposure to pollutants of indoor air and acute respiratory illness in children. The possibility of respiratory illness in children who live in a house, which has PM_{10} concentration 65 $\mu g/m^3$, were 7 times higher than that in children, who live in a house with PM_{10} level below 65 $\mu g/m^3$ (WHO, 2000). In China, researchers studied the relationship between wood or hay burning for cooking and asthma disease, which relationship was 39 % of women and 21% of men exposed occupationally (Xu et al., 1996). The exposure of indoor PM_{10} inclines adversely to the respiratory system including nasal pharyngeal and laryngeal, while indoor $PM_{2.5}$ can accumulate in the tracheobronchial region and alveoli. However, the surface of these particles can be adsorbed by organic pollutants (PAHs and metal) (Weisel, 2002). VOCs released from indoor have a strong association with mucous membrane irritation (Wolkoff et al., 2006; Cometto-Muñiz et al., 2004). In the US, indoor NO_2 exposure may also enhance asthmatic reactions to inhaled allergens, however it depends on poor ventilation, small room size and frequent use of gas stove for supplemental heating (Strand et al., 1997). Triche et al. (2005) reported that a 10 ppb

increasing of SO_2 was associated with increasing of wheezing and chest tightness, which were studied from the effect of indoor heating sources on respiratory illness in non- smoking women. Moreover, the main health effect of CO is a result of its ability to impair the oxygen binding capacity of hemoglobin, which is a cause of headaches, nausea, dizziness, breathlessness and fatigue, and with high exposures can lead to coma and death (Weaver et al., 2002). Furthermore, Burnett et al. (1997) found the strongest effect of indoor gaseous pollutants on elderly patients being hospitalized for congestive heart failure was CO, when compared with NO_2, SO_2 and O_3. Therefore, indoor air quality is an important issue that affects human health, which we spend most of our time indoors. There are many sources of indoor air pollutants emitted from different types, while amounts of pollutants depend on the material and fuels used as well as the type of human activity. Some of the causes of indoor air problems were found from poor ventilation systems, poorly designed residential and workplace buildings (Strand et al., 1997; Chunram et al., 2007; Klinmalee et al., 2009).

1.2 Gaseous pollutants; Nitrogen dioxide (NO_2)

Gaseous pollutants can have adverse effects on humans and the ecosystem. The pollutants can be emitted from natural origin or man-made. Pollutants are classified as primary or secondary. Primary pollutants are usually produced from a process such as volcanic eruption, motor vehicle exhaust and factories including carbon monoxide (CO), nitrogen oxide (NO_X) and sulfur dioxide (SO_2). Secondary pollutants are not emitted directly. Rather, they form in the air when primary pollutants react or interact such as ground-level ozone (O_3) and nitrogen dioxide (NO_2). Some pollutants may be both primary and secondary. The CO, SO_2, ground-level O_3 and NO_2 are the major source of gaseous pollutants in air quality index (AQI), as these pollutants can cause harm to human health and the environment and cause property damage.

EPA-National Ambient air Quality Standard has used nitrogen dioxide (NO_2) as the indicator for a larger group of nitrogen oxide (NO_X). NO_2 is the most toxic of nitrogen oxides and it is very importantly related to the human health risk. The concentration of NO_2 is well correlated with the concentration of CO (Masiol et al., 2012), and nitrous acid (HONO) and O_3 (Lee et al., 2002), SO_2, PAHs (Masiol et al.,

2012), PM_{10} (Xie et al., 2015) and $PM_{2.5}$ (Xie et al., 2015). Several studies investigated sources of NO_2 indoors. Miskinyte and Dedele (2015) measured NO_2 in seven homes in Kaunas city of Lithuania at different locations (bedroom, living room and kitchen). It was found that the average NO_2 were 20.1±4.46 μg/m^3 (bedroom), 23.1±5.02 μg/m^3 (living room) and 26.1±6.42 μg/m^3 (kitchen). They concluded that NO_2 was released from gas cookers. Arbox et al. (2007) found that gas stove cooking generates NO_2, while Ghosn et al., (2005) found that NO_2 released from butane gas heaters in residential homes (15.6-22.3 μg/m^3). Lewne et al., (2006) used personal exposure to monitor NO_2 level that drivers exposed to motor exhaust in a lorry, bus and taxi. NO_2 concentrations were 68 μg/m^3, 60 μg/m^3 and 48 μg/m^3, respectively. Moreover, the outdoor sources of NO_2 emitted from motor vehicles contributed to indoor NO_2 levels in houses and residential area closed to a main street (Cyrys et al., 2000). Jenkins et al. (2000) reported that indoor NO_2 levels emitted from tobacco smoking in homes were ranged from not detected to 76 ppb. Moreover, Concentrations of NO_X emitted from incense burning in a peak period (91±23 ppb and 90±7 ppb) at two temples were higher than non-peak period (48±10 ppb and 33±4 ppb) (Wang et al., 2007). International and national indoor and ambient air quality standards of NO_2 are shown in Table 1.1. Thailand standard for NO_2 levels in ambient air are 30 ppbv (1 hr) and 170 ppbv (annual average). The values are higher than other standard levels shown in Table 1.1. However, indoor air quality of NO_2 was announced by Environmental Sanitation Division of Bangkok Metropolitan administration (BMA) Bangkok, Thailand, in which 8 hours NO_2 concentrations in the workplace and residential area should not exceed 80 ppb (BMA, 2016).

Table 1.1 International Standard and guideline values for NO_2 indoors and in ambient air

Organization or country	Duration	Values			
		Indoor air		Ambient air	
		µg/m³	ppb	µg/m³	ppb
NAAQS/U.S.A	1 hour	-	100	-	100
	Annual	-	53	-	53
WHO	1 hour	200	106	200	106
	Annual	40	21	40	21
Canada	1 hour	-	-	-	53
	24 hours	170[a]	90[a]	-	213
	Annual	20[a]	11[a]	-	106
European	1 hour	-	-	200	106
	Annual	-	-	40	21
China	1 hour	-	-	200	106
	24 hours	-	-	80	42
	Annual	-	-	40	21
Hong Kong	1 hour	200	106	200	106
	8 hours	40 and 150[b]	< 21 and < 80[b]	-	-
	Annual	40	21	40	21
Singapore	1 hour	-	-	200	106
	Annual	-	-	40	21
Thailand[c]	1 hour	-	-	320	170
	8 hours	-	80	-	-
	Annual	-	-	57	30

Notes: NAAQS/EPA = National Ambient Air Quality Standard-U.S.-Environmental Protection Agency, 2006.

WHO = World Health Organization, 2005.

[a] = Residential Indoor Air Quality Guideline, Canada.

[b] = Indoor Air Quality Guideline Certification Scheme for Offices and Public Places; <21 ppb represents very good indoor air quality that a high-class and comfortable building should have and < 80 ppb represents indoor air quality that provides protection to the public at large including the very young and the aged.

[c] = Pollution Control Department (PCD) and Environmental Sanitation Division of Bangkok Metropolitan administration (BMA) Bangkok, Thailand

Inhalation of nitrogen dioxide is associated with respiratory irritant. There is evidence that indoor NO_2 is associated with an increase in respiratory symptoms among the general population, particularly in children. Previous studies investigated the relationship between indoor NO_2 exposure and respiratory symptoms in children with asthma. In the UK, they reported that indoor NO_2 was significantly associated with an increase in asthmatics symptoms at age ≤ 14 years, which related with the symptoms of chest tightness, breathlessness on exertion, daytime asthma attacks and night asthma attacks. Indoor NO_2 exposure at age 35-49 years was significantly associated with cough symptoms (Smith et al., 2000). Kattan et al. (2007) found that higher levels of indoor NO_2 in homes were associated with increased asthma symptoms between the age of 4-9 years and decreased peak flow, while asthmatic children between the age of 2-6 years were associated with high concentrations of indoor NO_2 in preschool inner-city children (Hansel et al., 2008). Moreover, Cibella et al. (2015) studied home exposure to high indoor NO_2 levels, which frequently occurs in adolescents living in southern Mediterranean urban area. It found that exposure of 40 ug/m^3 of indoor NO_2 or higher was significantly associated with the risk of increased frequency of respiratory symptoms and reduced lung function. In respiratory symptoms, the wheezes were act up higher than asthma, chronic and rhino conjunctivitis. However, indoor NO_2 exposure was associated with decreased lymphocytes in male and female and airway inflammation, which possibly is caused by the effects of exposure to NO_2 and could exacerbate airway disease and respiratory challenge (Frampton et al., 2002).

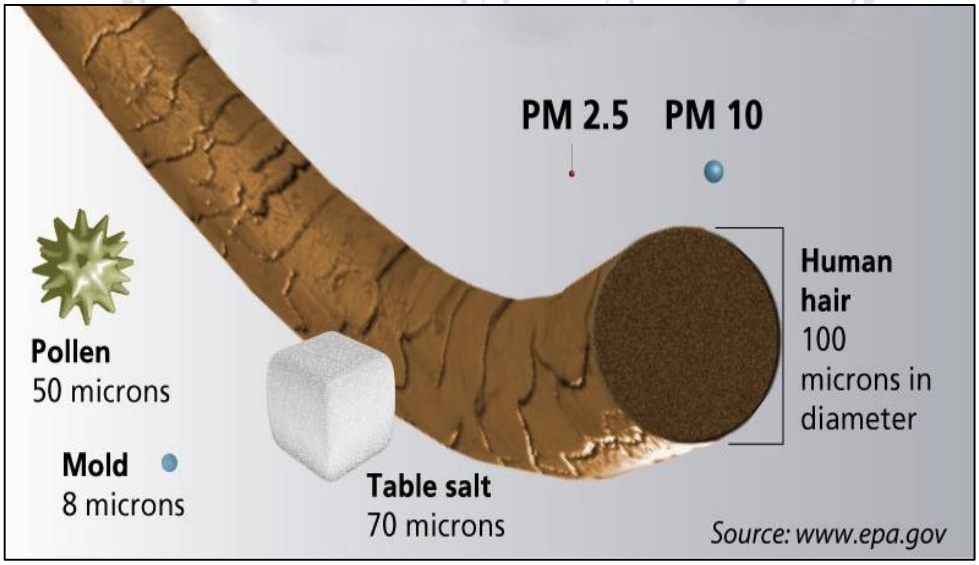
1.3 Particulate Matter (PM)

Particulate matters (PMs) are one of five pollutants in air quality index (AQI) and the most important in terms of adverse effects on human health. It is the term for particles and liquid droplets suspended in the air. Particulate matters are composed of both coarse particles (PM_{10}, particles with an aerodynamic diameter of 10 μm or less) and fine particles ($PM_{2.5}$, particles with an aerodynamic diameter of 2.5 μm or less) (WHO, 2005). The $PM_{2.5}$ and PM_{10} are often used as important indicators to characterize the mortality and health risk from lung cancer and cardiopulmonary deaths.

However, the mortality and health risk were stronger for $PM_{2.5}$ than PM_{10} (Fierro, 2000).

1.3.1 $PM_{2.5}$

Fine particles ($PM_{2.5}$) are particles that are less than 2.5 μm in diameter. There particles are less than approximately 1/30 to 1/40 the width of a human hair (60-100 μm) as shown in Figure 1.1. $PM_{2.5}$ is formed from gas and condensation of high-temperature vapors during combustion processes including forest fires or residential wood fires, burning of fossil fuels in motor vehicles, furnaces, boilers, and heaters and certain industrial processes. Secondary fine particles are created when chemicals react in the atmosphere and grow through particle-particle or gas-particle interactions. Sources of indoor $PM_{2.5}$ are traffic (from outside), biomass burning or wood stove, heater, tobacco smoking and incense burning.



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(Date: 19 Jan 2016)

Figure 1.1 Fine particulate matters ($PM_{2.5}$)

Table 1.2 Ambient air and indoor air quality standards of $PM_{2.5}$

Organization	Duration	Values	
		Indoor air	Ambient air
NAAQS/EPA	24 hours	35 μg/m^3	35 μg/m^3
	Annual	15 μg/m^3	15 μg/m^3
WHO	24 hours	-	25 μg/m^3
	Annual	-	10 μg/m^3
Canada	1 hour	100 μg/m^3	-
	Long term	40 μg/m^3	-
	24 hours	-	30 μg/m^3
EU	Annual	-	25 μg/m^3
Australia	24 hours	-	25 μg/m^3
	Annual	-	8 μg/m^3
China[a]	24 hours	-	75 μg/m^3
	Annual	-	35 μg/m^3
Japan	24 hours	-	35 μg/m^3
	Annual	-	15 μg/m^3
PCD, Thailand	24 hours	-	50 μg/m^3
	Annual	-	25 μg/m^3

Remarks and sources:

NAAQS/EPA = National Ambient Air Quality Standard-U.S.-Environmental Protection Agency, 2006.

WHO = World Health Organization, 2005.

EU = European Union

PCD = Pollution Control Department, Thailand

[a] = GB 3095-2012 ; class 2 standard apply to urban area

According to the study of Funk et al (2014), gases and particulate matters ($PM_{2.5}$ and PM_{10}) were investigated inside 628 residences of United Arab Emirates. They found that indoor $PM_{2.5}$ and PM_{10} levels were significantly correlated with vehicles parked with in five meters of the home, central air conditioning and having attached kitchens. Furthermore, $PM_{2.5}$ concentrations were investigated from inside 14 residential homes located in rural, roadside and urban homes. The $PM_{2.5}$ concentrations inside rural homes (173 $\mu g/m^3$) were higher than roadside homes (138 $\mu g/m^3$) and urban homes (136 $\mu g/m^3$). The major sources of roadside and urban homes were heavy traffic from cars, scooters and trucks throughout the day, while rural homes were influenced from smoking, usage of traditional stove with biomass fuel like wood, cow dung and kerosene and an improper ventilation system (Massey et al., 2009).

$PM_{2.5}$ concentrations were measured inside 28 venues of Lebanon and it was found that the major source emitted from secondhand tobacco smoking, which the mean $PM_{2.5}$ concentration was 342 $\mu g/m^3$ (Saade et al., 2010). Comparing with the study in Hungary (Tarnoki et al., 2009), where $PM_{2.5}$ samples emitted from tobacco smoke were measured in pubs, restaurants and café, the $PM_{2.5}$ level (103 $\mu g/m^3$) was lower than measured in Lebanon. In Thailand, Charoenca et al. (2013) revealed $PM_{2.5}$ levels emitted from smoking inside different types of venue were 488 $\mu g/m^3$ (bar), 131 $\mu g/m^3$ (restaurant), 58 $\mu g/m^3$ (transportation), 38 $\mu g/m^3$ (hospitals, hotels and office). Moreover, they found that the smoking was 2-4 times higher than smoke-free.

Titcombe and Simcik (2011) measured indoor $PM_{2.5}$ concentrations emitted from open wood fires, charcoal, a mix of charcoal and kerosene and liquid petroleum gas (LPG) for cooking fuels by using personal exposure sampler in households in the Njombe district of Tanzania. It was found that the average $PM_{2.5}$ from cooking in descending order were open wood fires (1,574±287 $\mu g/m^3$) > charcoal alone (588±347 $\mu g/m^3$) > kerosene/charcoal (88±42 $\mu g/m^3$) > LPG (14±3 $\mu g/m^3$). In Kuwait, indoor $PM_{2.5}$ levels were investigated in residential areas by using personal sampler. Average $PM_{2.5}$ concentrations emitted from cooking found kitchen (46-88 $\mu g/m^3$) were significantly higher than living room (22-61 $\mu g/m^3$) and bed room (24-32 $\mu g/m^3$) (Yassin et al., 2012).

Moreover, Fang et al. (2003) revealed $PM_{2.5}$ concentrations emitted from incense burning inside Taiwanese temples. Average $PM_{2.5}$ levels were 75.1 μg/m^3, while indoor $PM_{2.5}$ concentrations in the Hong Kong temple were reported as 230±15 to 360±23 μg/m^3 (Wang et al., 2007). The international standard values of $PM_{2.5}$ for indoor and ambient air contaminants are presented in Table 1.2.

$PM_{2.5}$ is composed of various combinations of ionic compounds (cations and anions), organic carbon (OC), metals and PAHs. Fang et al. (2003) found the median metallic element concentrations were Fe>Zn>Cr>Cd>Pb.>Mn>Ni>Cu in $PM_{2.5}$ from incense burning, while Ohura et al. (2005) revealed 88% benzo(a)pyrene (BaP) concentrations in $PM_{2.5}$ for inside residential homes in Japan. The dominant ion chemical components of $PM_{2.5}$ samples emitted from biomass burning were 78-85 % of nitrate (NO_3^-) and ammonium (NH_4^+) for indoor and outdoor environment, while K, Cl, S and Pb indicated the dominant of elements (Zhu et al., 2012). Furthermore, Hassanvand et al. (2015) showed that the predominant of PM-bound total PAHs was $PM_{2.5}$ (83-88 %) in outdoor and indoor a retirement and a school dormitory.

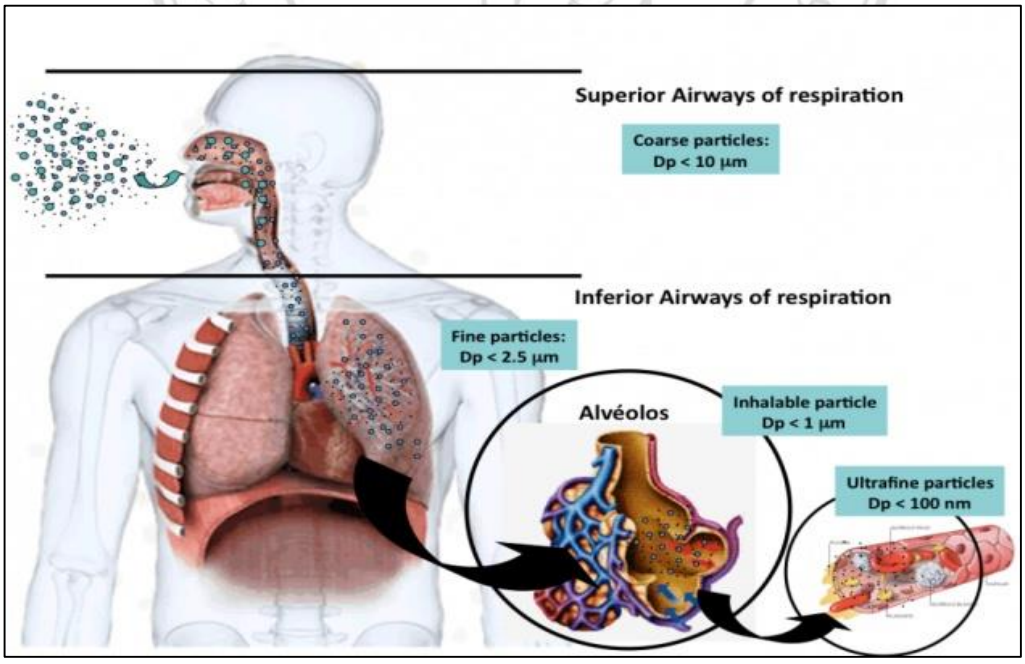

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Figure 1.2 Particles of various sizes to human respiratory system

Particle size is important for its impact on human health, which affects the respiratory system. The smaller the particle's size, the greater the particle's potential to cause adverse health effects (Figure 1.2). The study from Billet et al., 2007 found that about 92.15% of PM was $PM_{2.5}$, which collected from ambient in the heavy industrial city. Furthermore, PAHs are associated with particulate matter (PM). The predominant of PM-bound total PAHs found in $PM_{2.5}$ was 83-88 %, which can penetrate deep into the alveolar regions of the lungs (Hassanvand et al., 2015). Animeshkumaret et al. (1997) reported the particulate matter with aerosol dynamic diameter of 2 μm is 90 % respirable, whereas 2.5 μm is 75 % respirable. The increased $PM_{2.5}$ of each 10 $\mu g/m^3$ was associated with approximately a 4%, 6%, and 8% increase in all-cause, cardiopulmonary and lung cancer mortality respectively (Pope et al., 2002), while effects of increased 10 $\mu g/m^3$ $PM_{2.5}$ in 2 days associated with myocardial infraction, stroke and respiratory deaths were 1.18%, 1.78% and 1.68 %, respectively (Zanobetti and Schwartz, 2009). Pope et al. (2006) revealed ambient $PM_{2.5}$ of increase 10 $\mu g/m^3$ was associated with increased risk of acute ischemic coronary equal to 4.5 %. Fierro (2000) reported that $PM_{2.5}$ was associated with a 36 % increase in death from lung cancer and 26% in cardiopulmonary deaths and the risk being higher for people over the age of 65. The influence of increase 10.2 $\mu g/m^3$ $PM_{2.5}$ emissions from sugar cane burning was associated with a 21.4 % in child and elderly respiratory hospital admissions (Cancado et al., 2006). Cortez-Lugo et al. (2015) revealed the increased $PM_{2.5}$ of 10$\mu g/m^3$ was associated with approximately a 33 % and 23% increase in cough and phlegm in adults with chronic obstructive pulmonary disease (COPD), respectively. In eight southern European cities, increase of 10 $\mu g/m^3$ of $PM_{2.5}$ was associated with an increase in cardiovascular and respiratory admissions were 0.51 % and 1.36% (Stafoggia et al., 2013). Moreover, Caiazzo et al. (2013) used a multi-scale air quality model to apply and assess the health impacts of major combustion emissions of $PM_{2.5}$ in the U.S. It was indicated ~200,000 early deaths per year occur in the U.S. The $PM_{2.5}$ combustion related with mortalities in descending order were road transportation (~53,000 per year) > electric power generation (~52,000 per year) > industrial (~41,000 per year), due to the large surface area of $PM_{2.5}$, toxins, including polycyclic aromatic hydrocarbons (PAHs) and heavy metals are absorbed onto the surface. However, Billet et al. (2007) reported that the specific surface area of $PM_{2.5}$ was 1 m^2/g.

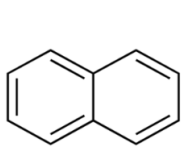
1.3.2 Polycyclic aromatic hydrocarbons (PAHs)

Combustion processes can generate PMs with carbonaceous cores that adsorb different organic toxicants (Mesquita et al., 2014), which are oxidized derivatives by photo-chemical oxidation process (Hallquist et al., 2009). Normally, fine particles are generated from fossil fuel and biomass burning combustion, while coarse particles are regularly generated from mechanical shearing and dust. Organic composition can be observed in both gaseous and particle phase including PAH, VOCs, persistent organic pollutants (POPs) and elemental carbon (EC) or black carbon (BC), etc. (Lee and Wang, 2004; Ohura et al., 2005; Hall et al., 2012; Qu et al., 2015). Moreover, the PMs size have been influenced by their organic composition and toxic potential (Ohura et al., 2006; Zhang et al., 2011; Qu et al., 2015). Especially, the strong toxic potential of submicron-sized particles was found in PAHs (Mesquita et al., 2014; Zhou et al., 2005).

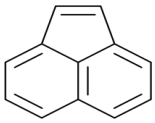
PAHs are comprised of a group of semi-volatile organic pollutants containing at least two fused aromatic rings. PAHs are widespread environmental pollutants, which originate mostly from incomplete combustion of fossil fuels and organic materials. Although some natural sources such as forest fires can contribute to the PAHs burden, human activities contribute most to PAHs emissions and the sources are entirely anthropogenic in urban and industrial atmospheres (Fang et al., 2002; Park et al., 2002; Vasilakos et al., 2007). The 16-PAHs in particulate matters emitted to ambient air have been determinated including naphthalene (NAP), acenaphthylene (ACY), acenaphthene (ACE), fluorene (FLU), phenanthrene (PHE), anthracene (ANT), fluoranthene (FLA), pyrene(PYR), benz[a]anthracene (BaA), chrysene(CHR), benzo[b]fluoranthene (BbF), benzo[k]fluoranthene(BkF), benzo[a]pyrene (BaP), indeno[1,2,3-cd]pyrene (IND), dibenz[a,h]anthracene(DbA), benzo[g,h,i]perylene (BPER), which are listed as the US EPA priority pollutants for the reference year 1995 (Villanyi, 2010). The structures of 16-PAHs are shown in Figure 1.3. Their physical and chemical properties are governed by size (number of carbon atoms) and shape (ring linkage pattern) of each molecule. A summary of some physicochemical properties and estimated carcinogenic potency are given in Table 1.3. The molecular weight (MW) of 16-PAHs was between 128 – 278 g/mol. The lowest MW was NAP while the highest

MW was DBA. Amount of PAHs were shown by water solubility values which was highest in NAP and lowest in IND. Boiling point (°C), log K_{OW} at 25 ºC and Henry's law constant (Pa m^3mol^{-1}) at 25 ºC.

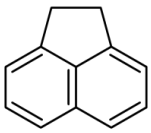
The carcinogenic potency was set up by the International Agency for Research on Cancer (IARC) and The United States Environment Protection Agency (U.S.-EPA). IARC classified 16-PAHs into 4 groups that consist of group 3: the compound is not classifiable as to being carcinogenic to human (ACE, FLU, PHE, ANT, FLA, PRY and BPER), group 2A = the compound is probably carcinogenic to human (DbA) and group 2B = the compound is possibly carcinogenic to human (NAP, BaA, CHR, BkF, BbF and IND) and group 1 = the compound is carcinogenic to humans (BaP). According to the classification by the U.S.-EPA, the 7-PAHs (BaA, CHR, BkF, BbF, BaP, IND and DbA) were grouped into the compound, which is probably carcinogenic (B2). NAP was grouped into the compound that is possibly carcinogenic to humans (group C), while ACY, ACE, FLU, PHE, ANT, FLA, PYR and BPER was grouped in to the group D and is not classifiable as to its carcinogenicity to humans.



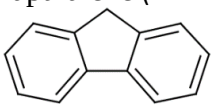
Napthalene (NAP)



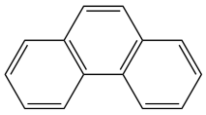
Acenaphthylen (ACY)



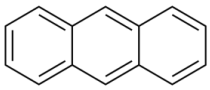
Acenaphthene (ACE)



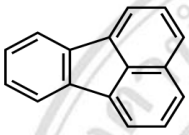
Fluorene (FLU)



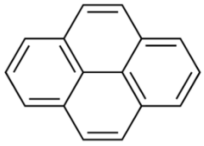
Phenanthrene (PHE)



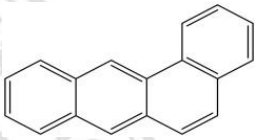
Anthracene (ANT)



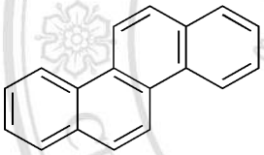
Fluoranthene (FLA)



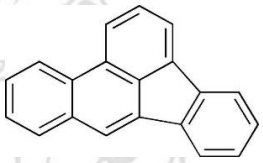
Pyrene (PYR)



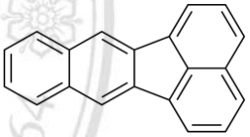
Benzo[a]anthracene (BaA)★



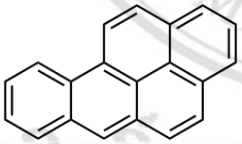
Chrysene (CHR)★



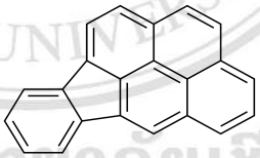
Benzo[b]fluoranthene (BbF)★



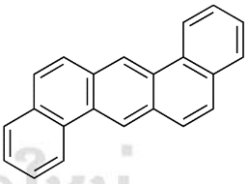
Benzo[k]fluoranthene (BkF)★



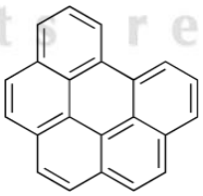
Benzo[a]pyrene (BaP)★



Indeno[1,2,3-c,d]pyrene (IND)★



Dibenzo[a,h]anthracene (DbA)★



Benzo[g,h,i]perylene (BPER)

★ = Carcinogenic PAHs

(Sources: US-EPA, 2008; IARC, 2010)

Figure 1.3 Structures of 16-PAHs

Table 1.3 Physical and chemical characteristics of US-EPA priority 16-PAHs

PAHs	Chemical formula	Molar mass (g/mol)	Water solubility[a] (mg/L)	Boiling point (ºC)	log Kow[a]	Henry's law constant[b] (Pa m^3/mol) at 25 ºC	Carcinogenic potency	
							IARC[c]	U.S.EPA[d]
NAP	$C_{10}H_8$	128	31	218	3.37	44.6	2B	C
ACY	$C_{12}H_8$	152	16	280	4	11.6	-	D
ACE	$C_{12}H_{10}$	154	3.8	279	3.92	18.5	3	D
FLU	$C_{13}H_{10}$	166	1.9	295	4.18	9.8	3	D
PHE	$C_{13}H_{10}$	178	1.1	340	4.57	4.29	3	D
ANT	$C_{14}H_{10}$	178	0.045	342	4.54	5.64	3	D
FLA	$C_{16}H_{10}$	202	0.26	384	5.22	1.96	3	D
PYR	$C_{16}H_{10}$	202	0.132	404	5.18	1.21	3	D
BaA	$C_{18}H_{12}$	228	0.011	437	5.91	1.22	2B	B2
CHR	$C_{18}H_{12}$	228	0.0015	448	5.91	0.53	2B	B2
BbF	$C_{20}H_{12}$	252	0.0015	481	5.8	0.051	2B	B2
BkF	$C_{20}H_{12}$	252	0.0008	480	6	0.044	2B	B2
BaP	$C_{20}H_{12}$	252	0.0038	495	5.91	0.034	1	B2
IND	$C_{22}H_{12}$	276	0.062	536	6.5	0.029	2B	B2
DbA	$C_{22}H_{14}$	278	0.0005	524	6.75	0.0075	2A	B2
BPER	$C_{22}H_{12}$	276	0.00026	>500	6.5	0.027	3	D

Remarks:

[a] = Stogiannidis and Laane (2015)

[b] = Shiu and Mackay (1997); IARC (2010)

[c] = IARC (2010)

[d] = US-EPA (2008)

PAHs are associated with particulate matter (PM) such as PM_{10}, $PM_{2.5}$ and $PM_{1.0}$. Evagelopoulos et al. (2010) collected $PM_{2.5}$ sample from urban areas surrounded by opencast coal mining emission and found that $PM_{2.5}$-bound PAHs concentrations were four times higher than PM_{10}-bound PAHs. The total PAHs on PM_{10}, $PM_{2.5}$ and PM_{10} was collected from indoor and outdoor at a retirement home and a school dormitory in Tehran. Moreover, Hassanvand et al. (2015) reported the highest percentage of carcinogenic PAHs bound to PM_{10}, $PM_{2.5}$ and $PM_{1.0}$ emitted from vehicles and natural gas combustion in descending order were $PM_{1.0}$ (49-56%) > $PM_{2.5}$ (37-43%) > PM_{10} (35-41%). Therefore, PAH's concentrations associated with particulate matters are highly dependent on fine particles (Duan et al., 2005; Billet et al., 2007; Hassanvand et al., 2015). Furthermore, Wu et al. (2014) reported a correlation between 18-PAHs in gas and particle phase and gas pollutants (NO_2, O_3, SO_2 and CO). They found that the particle phase PAHs (FLA, PYR, BaA, CHR, BbF, BkF, benzo[j]fluoranthene (BjF), benzo[e]pyrene (BeP), BaP, INP, DbA and BPER) were strongly correlated with NO_2 (r=0.648-0.842) and SO_2 (r=0.656-0.860), while gas phase PAHs was not a significant correlation (p<0.01). The $PM_{2.5}$-bound PAHs concentrations were correlated with SO_2 and CO (Evagelopoulos et al., 2010), while correlation between PM_{10}-bound PAHs and gas pollutants was found that CO and NO (Valerio et al., 2000). Moreover, the results of PAHs concentrations were significantly correlated with carbon dioxide (CO_2) inside urban areas and roadside homes, which was emitted from cooking and vehicles (Masih et al., 2010).

Several studies (Liao and Chiang, 2006; Chiang et al., 2009; Yang et al., 2015a; Zhang et al., 2015) used incremental life cancer risk (ILCRs) models to assess the human exposure to air borne-bound carcinogenic PAHs. They mentioned that the dermal contact ILCRs ranging from 10^{-6} to 10^{-4} is indicated a high potential cancer risk. Yang et al. (2015a) assessed the human health risk exposure to PAHs in 139 indoor dust samples that were collected from Guizhou province, southwest of China. The mean ILCRs inside the city, town, village and orefield were 6.14×10^{-6}, 5.00×10^{-6}, 3.08×10^{-6} and 6.02×10^{-6} for children and 5.92×10^{-6}, 4.83×10^{-6}, 2.97×10^{-6} and 5.81×10^{-6} for adults, respectively. Therefore, it was indicated as a high potential cancer risk. The city and town were exposed from industrial emission and vehicular exhausts, while the villages were mainly originated from cooking activity. The evaluation of inhalation exposure to

PM_{10}-bound PAHs of people was investigated at 8 night markets in urban area of a metropolis in eastern China by using incremental life cancer risk (ILCRs). The workers in the Hefei city night market were indicated as a potential cancer risk (1.13×10^{-3}) for exposure of 20 years. Moreover, they showed the probabilistic human health risk of worker exposed to PAHs on PM_{10} in night markets was 0.26% for 1 year, 66.0% for 3 years, 85.3% for 5 years, 93.7% for 10 years and 96.1% for 20 years. For customers, the maximum consumption time (t_{max}) for exposed PM_{10}-bound PAHs of the customer in night markets was found to be less than 1 hour for under 25 years old. Therefore, the assessment of the consumption time and acceptable ILCRs values depend on age (infants or toddlers) and children are vulnerable to high exposure risk by inhalation of PM_{10}-bound PAHs (Zhang et al., 2015). Qu et al. (2015) reported the incremental life cancer risk (ILCRs) values of $PM_{2.5}$-bound PAHs assessed the several environmental multi-pathways in Nanjing, China. The result of ILCRs values indicated that the potential cancer risk for adults was 2.62×10^{-5}, which the predominant of PAHs species were BaP, BbF and BaA. Furthermore, Hassanvand et al. (2015) revealed the predominant PM-bound PAHs concentrations found in $PM_{2.5}$ were 83-88 %, which can penetrate deep into the alveolar regions of the lungs. In *vitro* study, the metabolic activation of the very low doses of PAH-coated $PM_{2.5}$ onto the inorganic nuclei found that the underlying mechanism of action closely involved in its cytotoxicity in human lung epithelial cell (Billet et al., 2007). An epidemiological study found the relationship of traffic-related $PM_{2.5}$-bound PAHs and odd preterm birth was increased by 30 % per inter-quartile increase, which positively correlated and clustered together with the analysis (Wilhelm et al., 2011).

1.4 Indoor pollutant sources

The variety of combustion activities in an indoor environment includes cigarette smoking, biomass burning, incense and candle burning and stove (Bruce et al., 2000; Hall et al., 2012; Wiriya et al., 2015; Hassain, 2015). Moreover, the indoor environment was influenced from outdoor source including traffic and industries (Cao et al., 2005; Hassanvand et al., 2015; Bozkurt et al., 2015). These activities are considered to be major sources of indoor environment emissions of toxic pollutants.

1.4.1 Household

The source of the emissions of toxic pollutants in an indoor environment was studied by some researchers from selected households (Saade et al., 2010; Masih et al., 2012; Charoenca et al., 2013; Pokhrel et al., 2015). The major source of particles and gaseous pollutants for rural household in a developing country is found to be the burning of biomass for cooking with stove and heating with open fireplace, while an urban household is found to be smoking, traffic, using liquefied petroleum gas (LPG) for cooking, etc.. Moreover, incense and candle burning is the major source of indoor environment at households in many countries such as Thailand, Hong Kong, China, India and Taiwan (Li and Ro, 2000; Lee and Wang, 2004; Lin et al., 2008; Yang et al., 2007b; Orecchio, 2011).

1) Biomass burning (stove and heating)

The 2.6 billion people in 2015 of the rural areas in developing countries use biomass for their energy needs such as for cooking using wood fuel, charcoal, animal dung and agricultural residues, which burning of biomass will increase to be over 2.7 billion by 2030. These resources are higher than 90% of household energy consumption in many countries (IEA, 2006). The toxic pollutants emitted from biomass burning are CO, NO_X, SO_2, PMs and organic composition including PAHs and VOCs. Masih et al. (2012) investigated the source of particle-bound PAHs in indoor urban residential and roadside homes in Northern India. It was found that they were emitted from cooking, tobacco smoking and incense burning. Furthermore, household activities, which rely on cooking on a stove, indoor smoking, outdoor vehicular traffic and garbage burning were found to be the major source of PM_{10}, $PM_{5.0}$, $PM_{2.5}$ and $PM_{1.0}$ (Massey et al., 2012), while Shen et al. (2013) reported PAHs bounded PMs emitted from domestic stoves in rural China was found that 80% of $PM_{2.5}$-bound PAHs and 65% for $PM_{1.0}$-bound PAHs. Chakraborty et al. (2014) reported emission factors (EFs) of CO, CO_2 and O_3 emitted from biomass fuel (dung, straw, wood, leaf, coal cake, twigs and gas stove) in rural areas in West Bengal, India. It was found that the concentrations of gaseous pollutants in a descending order were $CO_2 > O_3 > CO$, while Pokhrel et al. (2015) showed $PM_{2.5}$ levels emitted from fuel stove type in a descending order were

biomass > Kerosene > LPG > electric. Moreover, Wiriya et al. (2015) revealed the EFs of rice straw, maize residues and leaf litter burning in Chiang Mai, Thailand found the highest PM_{10} and PM_{10}-bound PAHs was emitted from leaf litter. Exposure to biomass smoke indoors may be associated with many adverse health effects including respiratory illness, chronic pulmonary disease, lung cancer and lung irritation, asthma and pulmonary tuberculosis (Garrett et al., 1999: Bruce et al., 2000). Thus, indoor combustion of biomass fuels on stove and heating is one that has affected human health risk for cause of death worldwide.

2) Tobacco smoking

Tobacco smoke is a cause of morbidity and mortality that contributes to human health indoor environment. Due to tobacco smoke, it is a complex carbon-based, dynamic aerosol suspended in an equally complex organic vapor mixture (Ghosh and Ionita, 2007). The toxic pollutants generated from tobacco smoking include nicotine, CO, SO_2, NO_2, PMs ($PM_{2.5}$&PM_{10}), PAHs. In the previous studies, Saade et al. (2010) found that $PM_{2.5}$ concentrations were released from indoor second-hand tobacco smoke (cigarette and water-pipe smoking) at a venue in six Lebanese cities. Charoenca et al. (2013) revealed that $PM_{2.5}$ levels emitted from smoking inside venue types in Thailand in a descending order were bar > restaurant > transportation > hospitals, hotels and office. Moreover, Cigarette smoke can release benzene and PM_{10} concentrations (Tirler and Settima, 2015). Moreover, Matt et al. (2004) reported measurement of nicotine in infant urine (≤1 year) emitted from smoking and non-smoking. It was found that infant exposure inside households were 5-7 times higher than those non-smokers. In the USA. The U.S.-Department of Health and Human Services reported 5.6 million of todays' children will ultimately die early from exposure to smoking and 1 out of 3 cancer deaths in this country could be smoking (US-HHS, 2015). However, tobacco smoke can be harmful causing many diseases and deaths including the trigger to an asthma attack, affecting the heart and blood vessels, increasing risk of heart attack and stroke in non-smokers and lung cancer (Janerich et al., 1990; Thomas et al., 2014; Jung et al., 2015).

3) Incense burning

Incense burning is one general human activity inside the home in many countries, which has been associated with many culture and religions. The gaseous pollutants and particulate matter are contaminated incense smoke, which may be adsorbed as toxic pollutants including CO, NO_X, SO_2, VOCs, PAHs and toxic metals. Li and Ro (2000) reported particles-bound PAHs concentrations inside a Taiwanese home, which the burned incense was 158 times higher than non-burned incense. Moreover, Cao et al. (2005) found $PM_{2.5}$, organic carbon (OC) and elemental carbon (EC) released from incense burning at residential homes in Hong Kong. The inside homes of the United Arab Emirates, incense smoke contributed PMs, CO, NO_X and formaldehyde (HCHO) to indoor environment (Cohen et al., 2013). An indoor experimental house with incense burning, Ji et al. (2010) showed $PM_{2.5}$ levels in the living room were higher than the kitchen, while CO_2 concentrations in a descending order were living room > kitchen > bedroom. Moreover, Pan et al. (2014) revealed long-term exposure to incense burning in homes was associated with an increased risk of cardiovascular mortality, while Wang et al. (2011) reported incense burning at home was associated with an increased risk of asthma and wheezing. However, incense burning is a popular practice inside temples more than residential homes and the workplace, as described in Section 1.5.2.

1.4.2 Worship places (Shrine)

Several studies inside worship places, the temple, shrine and church have been conducted worldwide to determine gaseous pollutants and particulate matters with adsorbed organic pollutants. The major sources of toxic pollutants were incense and candle burning inside worship places. (Huynh et al., 1991; Fine et al., 1999; Liao and Chiang et al., 2006; Wang et al., 2007; Wu et al., 2009; Derudi et al., 2012 and 2014; Dewangan et al., 2014). Incense and candle in the worship places are an important religious tradition and ritual activity in many countries and societies.

Asian countries, where mainstream Confucianism, Buddhism and Taoism religions such as China, Hong Kong, Thailand, India and Taiwan, incense and candle burning is a common practice as an important religious ceremony in all temples and shrines. In Thailand, the practice of shrines can be found to be using an amount of

incense more than in temples because one of the reasons is that there are many altars inside the shrine. Especially, Chinese folk temples and shrines were established by Thai of Chinese origin ancestors, which had religious tradition and ritual activity and follow the Chinese lunar calendar. During Chinese New Year or other festivals (Chinese Ghost festival, the moon festival, a vegetarian festival), the many pilgrims and visitors will observe and pay respect to Buddha and other deities when a huge amount of incense and candle is burned in temples and shrines. The consumption of incense and candle may even double or triple that estimated amount and it may indicate an environmental hazardous situation in the indoor environment.

1) Candle burning

One of main indoor environment sources is candle burning. Candle burning is a general practice for religious tradition in a temple or shrine. Moreover, the burning of scented candles and aromatherapy candles are burned and decorated in households (Derudi et al., 2012). The main ingredient of candle manufacturing is paraffin wax, which is made from petroleum derivative (Orecchio, 2011; Derudi et al., 2012) and the natural ingredients are bees wax and stearin made from animal and vegetable oil (Guzialowska-Tic, 2013). When a candle is burned, it releases gaseous pollutants, particular matters and trace of organic pollutants (Fine et al., 1999; Orecchio, 2011; Derudi et al., 2012). Emission factors (EFs) of candle burning on particulate matter concentrations from a chamber in Hong Kong was studied by Lee and Wang (2007). It was found that the range of $PM_{2.5}$ and PM_{10} concentrations was 19 to 44 μg/m^3 and 22 to 54 μg/m^3, respectively and $PM_{2.5}/PM_{10}$ ratios ranged 78-91 %. Moreover, candle burning emitted gaseous pollutants, including CO, NO, NO_2 NO_X, methane (CH_4), NMHC (non-methane hydrocarbons) and VOCs. These toxic pollutants emitted from different ingredients of candle production in a descending order were gel wax > paraffin wax > bees wax. Moreover, Fine et al. (1999) reported $PM_{2.5}$ concentrations emitted from paraffin (0.52-3.72 mg/g) were higher than beeswax (1.46-2.04 mg/g). Derudi et al. (2012) reveals EFs of aldehydes (formaldehyde, Acetaldehyde, proplonaldehyde, butyraldehyde and benzaldehyde) emitted from candle burning in the test chamber. The highest EFs of evidenced candles was found to be formaldehyde.

Moreover, this study has confirmed the candle made from pure paraffin to emit aldehydes was less than dye-scented candles.

In Italy, Orecchio (2011) investigated 18-PAHs emitted from decorative candles in an indoor environment. The concentrations of total PAHs emission factors ranged from 2.3 to 50.0 µg/kg and the carcinogenic PAHs concentration ranged from 0.09 to 2.00 µg/kg. The predominant PAHs were NAP, ACE, ACY and FLU. In 2014, they reported the highest EFs of PMs emitted from candle burning, which was $PM_{0.25}$ (2-239 mg/g). The EFs of total PAHs and carcinogenic PAHs concentrations were 27-152 ng/g and 0.1-11 ng/g, respectively. Moreover, candle burning released EFs of BTEX, CO, NO_X and SO_2 were 0.12 to 1.10 µg/g, 2.44±0.07 to 3.99±0.05 mg/g, 0.87±0.03 to 0.92±0.03 mg/g and 1.07±0.24 to 5.52±0.30 mg/g, respectively (Derudi et al., 2014). Therefore, exposure to candle smoke is a concern for the potential health effects from toxic pollutants emission such as PAHs, aldehyde and gaseous pollutants.

2) Incense burning

Incense burning has been a common practice for many countries. In Christian countries, incense burning was used in the Eucharistic ceremony in churches which symbolized the ascent of the prayer of faithful and merit of the saint. After that, incense was used continuously in Catholic Christendom churches which its employ during divine worship (Lin et al., 2008). Asian countries where are mainstream Buddhism and Taoism religions such as China, Thailand and Taiwan. Incense burning is a daily practice and worship religious ceremonies and also a tradition to pay respect to ancestors (Estrellan and Iino, 2010). Furthermore, incense and joss paper burning has been an important ceremonial practice for deity worshipping in these religions especially China and Taiwan.

There are several forms of incense including stick, cone, coil, powders, rope and rocks or charcoal (Jetter et al., 2002). Incense sticks are popular in worshiping. Incense sticks made in Asian countries which various ingredients used to produce incense such as wood powders, herb, woody plant, spices, plant-based glutinous powders, seed, root, flowers, essential oils, synthetic substitute chemicals

which are used in the perfume industry and water (Chang et al., 2007; Jetter et al., 2002).

Burning incense emits a lot of gaseous and particulate pollutants (Lee and Wang, 2004; Lin et al., 2008; Yang et al., 2007a). For example, incense was found to be a significant source of PAHs, heavy metals, particulate matters (TSP, PM_{10} and $PM_{2.5}$), greenhouse gases (CO_2 and CH_4) and volatile organic compounds (VOCs). Exposure to incense smoke may be associated with many adverse health effects including cancer, respiratory morbidity, central nervous system damage and lung irritation (Chiang and Liao, 2006; Lin et al., 2008; Navasumrit et al., 2008).

1.5 Literature review on emission of incense burning

According to the results of several studies, incense burning is a significant source of particulate matters (PMs), metal elements and PAHs, gaseous pollutants (CO, CO_2, SO_2 and NO_X) in the household and temple. Therefore, the impact of incense burning is also an important issue to be investigated.

1.5.1 Incense burning in worship places

The effect of incense burning on particulate matter concentrations from the Tzu Yun Yen temple in central Taiwan was studied by Fang et al. (2002). The analysis of concentrations for indoor suspended particulate ($PM_{2.5}$, $PM_{2.5\text{-}10}$ and PM_{10}) increased during the time when pilgrims appeared at the incense burning period. The average $PM_{2.5}$, $PM_{2.5\text{-}10}$ and PM_{10} concentrations were 63.3, 26.5, 89.7 $\mu g/m^3$, respectively. The $PM_{2.5}/PM_{10}$ ratios ranged between 31.2-87.4 % and averaged 69.6 ± 12.3 % during the incense burning period. Moreover, burning incense was found to be a significant source of various pollutants. Lung and Kao (2003) reported the emissions of $PM_{2.5}$ and PM_{10} concentrations released from two temples in Taiwan. They found that the concentrations $PM_{2.5}$ (330±1.9 $\mu g/m^3$ to 610±1.4 $\mu g/m^3$) and PM_{10} (539±1.5 $\mu g/m^3$ to 626±1.4 $\mu g/m^3$) were 5-10 and 6-7 times higher than studied by Fang et al. (2002), respectively. The ratios of $PM_{2.5}/PM_{10}$ ranged 54-63 %, where the values were higher than the ranged values (32.6-65.0 %) found by Wu et al. (2015). However, Wu et al.

(2015) studied the emission of $PM_{2.5}$ and PM_{10} during the peak tourist season at Mount Wutai Buddhism, Shanxi province, China. The $PM_{2.5}$ and PM_{10} concentrations were 1.43 to 59.20 μg/m^3 and 17.40 to 161.45 μg/m^3, respectively. The main source was released from incense and candle burning and accompanied coal burning from villages. The characteristic emissions of air pollutants from incense burning in a large environmental test chamber in a study by Lee and Wang (2004), found three types of incense and were compared in the context of tradition, aromatic and church incense. The target pollutant included particulate matters (PM_{10} and $PM_{2.5}$), volatile organic compounds (VOCs), carbon monoxide (CO), carbon dioxide (CO_2), nitrogen oxide (NO_X) and methane (CH_4). The particulate matter and CO emitted from all the incense significantly exceeded the Recommended Indoor Air Quality objectives for Office Building and Public Places in Hong Kong (HKIAQO). It was observed that when comparing the gas pollutant emission factors between two major incense categories (traditional and aromatic), generally, the VOCs emitted sequence was aromatic incense> traditional incense> church incense. The results showed that incense burning was one of the important indoor air pollution sources for PM, CO and VOCs. Cheng et al. (2015) measured VOCs emitted from sandalwood incense coils in a chamber. The VOCs concentrations were collected on a needle trap sampler (NTS) packed with 60-80 and 100-120 mesh divinylbenzene (DVB) adsorbents and analyzed by GC-FID. They found that BTEX (Benzene, Toluene, Ethylbenzene and o-Xylene) values were 2,285 ng/mg and 4,481 ng/mg for 60-80 and 100-120 mesh DVB-NTS, respectively. Yang et al. (2007b) studied particle size distribution and concentrations of PAHs in incense smoke in a custom-designed combustion chamber. The total toxins of the benzo[a]pyrene equivalent concentration of the solid-phase PAHs (S-PAHs) was over 40 times higher than that of the corresponding gas-phase PAHs, indicating that the S-PAHs in incense smoke may pose potential health risk.

In Taiwan, Lin et al. (2001) studied total suspended particulate (TSP) and particulate - bound total PAH concentrations (21 PAHs) from inside and outside of a Taiwanese temple. PAH concentrations found in the inside air of the temple ranged from 4,739 – 11,130 ng/m^3 with the average concentration of 8,888 ± 3,598 ng/m^3. The average value of total-PAH concentration was approximately 19 times higher than that found outside the temple (468 ±156 ng/m^3). The TSP concentrations in the indoor were

between 429-991 μg/m^3 and average 790 μg/m^3. The average TSP concentration was approximately 3-11 times higher than outdoor of the temple (72.5 μg/m^3). In 2002, they reported the average total suspended particle – bound PAHs concentrations of 6,258 ng/m^3 and 231 ng/m^3 in the indoor and outdoor air of a Taiwanese temple, respectively. The PAH concentrations of the temple's indoor air were 27 times higher than those of outdoor air. The highest top five PAH concentrations (particulate + gas phase) were identified as 3,583 ng/m^3 acenaphthylene, 1,264 ng/m^3 naphthalene, 349 ng/m^3 acenaphthene, 243 ng/m^3 fluoranthene and 181 ng/m^3 phenanthrene (Lin et al., 2002). Liao and Chiang (2006) measured particle-bound PAHs and BaP concentrations emitted from incense burning inside and outside a Taiwanese temple. It found that median values indoor particles-bound PAHs and BaP were 478 ng/m^3 and 101.8 ng/m^3, while concentrations of outdoor were 232 ng/ m^3 and 154 ng/m^3, respectively. However, the median concentrations of outdoor BaP were higher than indoor values because the outside was influenced from emission of traffic density. In India, Dewangan et al. (2014) investigated the emission factor of particle-bound PAHs released from ritual based burning activities inside different Indian religious places (Marriage Places, Muslim Holy shrines, Buddhist temples, Hindu temples and Cremation center). The results showed that the highest emission factor of total-PAHs was found at Muslim Holy shrine (99.09±6.02 mg/kg) and Buddhist temples (90.35±6.76 mg/kg). The major source came from incense, candle and styrax benzoin burning activities. Moreover, they found that the dominant PAHs rings were 4 and 5 rings from all religious and ritual sites.

Li and Ro (2000) found incense burning is associated with increased level of PAHs in homes. The investigated home in their study with incense burning did have higher concentrations of FLA (fluoranthene), PYR (pyrene), BaP (benzo[a]pyrene) and BPER (benzo[g,h,i]perylene). Lung et al. (2003) investigated indoor PM_{10} concentrations resulting from incense burning in household environments under two conditions (closed and ventilated). The exposure concentrations of particle-bound PAHs (sum 13 PAHs) were estimated. The results showed the incense burning of indoor PM_{10} and particle-PAHs concentrations in a closed condition (326-723 μg/m^3 and 0.20-0.45 μg/m^3) were significantly higher than those of a ventilated condition (109-178 μg/m^3,

0.088-0.11 μg/m^3). Concentrations were elevated for at least 6 hours under closed conditions.

Numerous studies related to inhalation epidemiology have shown a positive association between ambient PMs - bound PAHs concentrations and adverse health effects. Navasumrit et al. (2008) reported that incense smoke emitted benzene, 1, 3-butadiene and polycyclic aromatic hydrocarbons (PAHs) from a temple in Samutprakarn, Chachengsao and Ayutthaya Provinces of Thailand. Samples were collected by using personal air sampling (thermal desorption tube) and analyzed by GC-MS. They were compared between temple workers (n= 40) and control workers (n= 25). It was found that temple workers were exposed to a relatively high level of benzene (45.90 μg/m^3), 1, 3-butadiene (11.29 μg/m^3) and PAHs (19.56 ng/ m^3), which were significantly higher than those of control workers ($p<0.01$). These results indicated that exposure to carcinogens emitted from incense burning may increase the health risk for the development of cancer in temple workers. In Taiwan, the exposure of the temple workers was investigated and PM_{10} and PAHs concentrations were collected by using personal collection samplers during pilgrim days (the first and fifteenth day of each lunar month) and normal days (all other days). Concentrations of total PAHs (t-PAHs) and carcinogenic PAHs (car-PAHs) on pilgrim days were found to be higher than those on normal days. Mean concentrations of urinary 1-hydroxypyrene (1-OHP) in the pre-shift and post-shift workers of the temples on normal days were 1.20 and 1.61 μg/g creatinine, respectively. However, the post-shift concentrations of 1-OHP found in the temple workers on pilgrim days were all higher than those on normal days. In other words, an increase of unit concentration of urinary 1-OHP would lead to an approximate 32-fold increase of cancer risk for the workers of the temples (Kuo et al., 2008). Chiang and Liao (2006) reported the assessment of human exposure to airborne PMs and PAHs from heavy incense burning in temples using a mechanistic-based exposure and risk models, appraised with reported empirical data. Results implicate that exposure to smoke emitted from heavy incense burning may promote a lung cancer risk. Moreover, Pan et al (2014) presented a study of epidemiologic evidence that long-term exposure to indoor air pollution from incense burning substantially contributed to the risk of cardiovascular mortality at a population level (45-74 age years) in Singapore. They found that 76.9 % of current incense users, whose is daily incense burning for ≥

20 years was a 12 % higher risk of cardiovascular mortality. Moreover, incense burning was associated with approximately 8% of coronary heart disease (CHD) deaths and 12% of stroke deaths. Hwang et al. (2014) studied the correlation between the frequency of incense burning at home and the blood lead level of children (<6 years). The result was not significantly correlated. However, the frequency of incense burning at home was significantly correlated with increased metals in blood levels (p=0.0022), which metals in blood level in descending order were zinc (Zn) > copper (Cu) > Lead (Pb) > manganese (Mn) > mercury (Hg) > arsenic (As). Furthermore, human exposure to airborne- bound carcinogenic PAHs emitted from incense burning is assessed by using incremental life cancer risk (ILCRs) models, which the acceptable ILCRs range from 10^{-6} to 10^{-4} indicates a high potential cancer risk. Chiang et al. (2009) investigated the assessment of probabilistic human health risk exposed to carcinogenic PAHs in particles emitted from incense burning for temple goers/workers in a Taiwanese temple. A 95% probability total ILCR (9.87×10^{-4} to 1.13×10^{-3}) of workers extremely exposed to carcinogenic PAHs in the temple were indicated to be of a high potential health risk. The assessment of probabilistic human health risk exposed to carcinogenic PAHs for a child (5-7 years), adolescent (17-23 years) and adult (48-65 years), While found that 90% probability of ILCR (10^{-7} to 10^{-6}) indicated an unacceptable human health risk based on average 3 hours residence time (Liao and Chiang, 2006). However, inhalation of PAHs of which some are considered carcinogens in particulates, is a potentially serious health risk linked to an excess risk of lung cancer.

1.5.2 Incense burning in chamber for pollutant emission study

The particle-bound PAHs emitted from incense burning in the chamber has been investigated in some previous studies (Lung and Hu, 2003, Jetter et al., 2002 and Yang et al., 2012a; Kuo and Sopajaree, 2016). The emission factors of fine particulates ($PM_{2.5}$) concentrations from incense burning in a combustion chamber was studied by Jetter et al. (2002). The $PM_{2.5}$ emission factors emitted from incense burning was 5-56 mg/g which was higher than that from See and Balasubramanian (2011) (0.4 ± 0.0 to 44.5 ± 5.0 mg/g). Lung and Hu (2003) studied the emission factors of $PM_{2.5}$ and PM_{10} concentrations from incense burning in a chamber. They found that the PM_{10} emission factors (18.9±1.03 to 42.7±1.10 mg/g) were lower than the emission factors of

$PM_{2.5}$ concentrations (20.8±1.10 to 44.5±1.03 mg/g). Kuo and Sopajaree (2016) revealed the emission factors of $PM_{2.5}$ emitted from four brands of traditional incense stick (Taiwan and Thailand) in chamber. They found that $PM_{2.5}$ emission factors from producing Taiwan (11.37±0.73 and 23.38±2.02 mg/g) was higher than in Thailand (11.08±1.27 and 13.93±0.61 mg/g). The emission factors of particle- bound PAHs concentrations ranged from 17.1 to 25.2 µg/g, while PHE, ACE and FLA were the major species of PAHs. Solid and gas-phase PAHs emission factors from burning of nine types of incense were 6.0 ± 0.73 µg/g and 17.6 ± 5.7 µg/g, respectively (Yang et al.,2007a). The relationship between particulate and PAHs from burning various types of incense in a sampling chamber was studied. The emission factors for total particulate mass were 4.19-82.16 mg/g, while those of PAHs were 1.20-9.50 µg/g, respectively (Yang et al., 2012a).

The materials made in an incense stick were investigated in Singapore by See and Balasubramanian (2011). They studied the emission of fine particle ($PM_{2.5}$) from different incense burning in a chamber such as agar wood, cedar wood, sandalwood and rose incense stick. It appeared that the highest $PM_{2.5}$ concentrations emitted from rose incense burning. Cedar wood and agar wood emitted a higher $PM_{2.5}$ than sandalwood. Furthermore, the use of essential oil as an additive could have led to increased emissions of $PM_{2.5}$, which may explain why rose incense had the highest emission rate. Yang et al. (2012a) revealed the emission factor of particle-bound PAHs emitted from the burning of several brands of incense in the chamber. It found that the highest total-PAHs concentrations was white sandalwood brand (9.50±0.93 µg/g), which it was produced from white sandalwood powders and tree bark powder. The lowest emission factor was smoke-free incense (1.20±0.04 µg/g). Furthermore, the result of predominant PAHs species were indicated to be FLA, PHE, PYR and CHR. The emission factors of particle-bound PAHs concentrations from sandalwood incense burning in a combustion chamber was studied by Yang et al. (2012a). The total-PAHs emission factors emitted from incense burning was 6.71±0.23 to 9.50±0.93 µg/g, which was lower than that of agar wood incense burning from Lung and Hu (2003) (17.1±1.2 to 25.2±1.2 µg/g). Sawdust, a major component of incense, was burned in an aluminum foil-lined wall chamber. The emission rates ranged from 154-255 mg/h of $PM_{2.5}$ and 378-790 mg/h of CO, while the $PM_{2.5}$ and CO emission factors were 65.6-252 mg/g and

242-454 mg/g, respectively (Ongwandee and Pipitthakul, 2010). The emission of benzene and PM_{10} concentrations from vanilla, cedar, traditional and cinnamon incense in an indoor room were investigated by Tirler and Settimo (2015). The results of the highest PM_{10} and benzene concentrations emitted from traditional incense were 339 $\mu g/m^3$ and 205 $\mu g/m^3$, respectively. The values of PM_{10} in descending order were vanilla incense (342 $\mu g/m^3$) > cedar incense (247 $\mu g/m^3$) > cinnamon incense (211 $\mu g/m^3$), while benzene concentration was found with cedar incense (53 $\mu g/m^3$) > vanilla incense (42 $\mu g/m^3$) > cinnamon incense (12 $\mu g/m^3$).

There are limited data of pollutants emitted from incense burning in Thailand. Incense burning is a serious issue concerning human health effect. The purpose of this study has been to provide reliable data and information concerning incense burning and its pollutant emissions.

1.6 Research objectives

1.6.1 To determine air pollutants both in forms of particulate ($PM_{2.5}$ and $PM_{2.5}$-bound PAHs) and some toxic gases emitted from the burning of different incense types in a combustion chamber.

1.6.2 To investigate pollutant profiles and emission factors of pollutants from the burning of different incense types.

1.6.3 To measure air pollutants emitted in a real environment from incense burning in selected shrines.